PB82-207127

Impacts on Human Health from the Coal and Nuclear Fuel Cycles and Other Technologies Associated with Electric Power Generation and Transmission

Ohio River Basin Energy Study
Urbana, IL

Prepared for

Environmental Protection Agency
Washington, DC

Jul 80

U.S. Department of Commerce
National Technical Information Service
NTIS

TECHNICAL REPORT DATA
(Please read Instructions on the reverse before completing)

1. REPORT NO.	2.	3. RECIPIENT'S ACCESSION NO.
EPA-600/7-81-055		PB82-207127

4. TITLE AND SUBTITLE	5. REPORT DATE
Impacts on Human Health from the Coal and Nuclear Fuel Cycles and Other Technologies Associated with Electric Power Generation and Transmission	July 1980
	6. PERFORMING ORGANIZATION CODE
7. AUTHOR(S)	8. PERFORMING ORGANIZATION REPORT NO.
Edward P. Radford, M.D.	
9. PERFORMING ORGANIZATION NAME AND ADDRESS	10. PROGRAM ELEMENT NO.
University of Pittsburgh Pittsburgh, Pennsylvania 15261	C2JN1E
	11. CONTRACT/GRANT NO.
	Subcontract under Prime Contract R805588
12. SPONSORING AGENCY NAME AND ADDRESS	13. TYPE OF REPORT AND PERIOD COVERED
U.S. Environmental Protection Agency Office of Research and Development Office of Environmental Engineering and Technology Washington, DC 20460	14. SPONSORING AGENCY CODE EPA/600/17

15. SUPPLEMENTARY NOTES

This project is part of the EPA planned and coordinated Federal Interagency Energy/Environment Research and Development Program.

16. ABSTRACT

This report was prepared in support of the Ohio River Basin Energy Study (ORBES), a multi-disciplinary research program supported by the Environmental Protection Agency. It evaluates major public health impacts of electric power generation and transmission associated with the nuclear fuel cycle and with coal use. Only existing technology is evaluated. For the nuclear cycle, effects of future use of fuel reprocessing and long-term radioactive waste disposal are briefly considered. The health effects of concern are those leading to definable human disease and injury. Subjective factors, effects of pollutants on visibility, and effects on agriculture and aquatic life forms are outside the scope of this report.

Health effects are scaled to numbers of persons and activities associated with a nominal 1000-megawatt electric plant fueled by either option. Comparison of the total health effects to the general public shows that the health risks from the coal cycle are about 50 times greater than for the nuclear cycle (coal, 0.7-3.7 major health effects per 1000 MWe per year; nuclear, 0.03-0.05 per 1000 MWe per year). For workers, these rates are higher (coal, 6.5-10.9 major health effects per 1000 MWe per year; nuclear, 4.6-5.1 per 1000 MWe per year). No evidence is found that electrical transmission contributes any health effects to the general public, except when broken power lines come in contact with people.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
Environment Energy Conversion	Health Effects Integrated Assessment Extraction Processing Utilization Coal Nuclear	

18. DISTRIBUTION STATEMENT	19. SECURITY CLASS *(This Report)*	21. NO. OF PAGES
Release to the Public	Unclassified	58
	20. SECURITY CLASS *(This page)*	22. PRICE
	Unclassified	

EPA Form 2220-1 (Rev. 4-77) PREVIOUS EDITION IS OBSOLETE

PB82-207127

EPA-600/7-81-055
July 1980

IMPACTS ON HUMAN HEALTH FROM THE COAL AND NUCLEAR FUEL CYCLES AND OTHER TECHNOLOGIES ASSOCIATED WITH ELECTRIC POWER GENERATION AND TRANSMISSION

by

Edward P. Radford, M.D.
University of Pittsburgh
Pittsburgh, Pennsylvania 15261

Prepared for

Ohio River Basin Energy Study (ORBES)
Subcontract under Prime Contract R805588

Project Officer

Lowell Smith
Office of Environmental Engineering and Technology
Office of Research and Development
U.S. Environmental Protection Agency
Washington, D.C. 20460

OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
WASHINGTON, D.C. 20460

REPRODUCED BY
NATIONAL TECHNICAL
INFORMATION SERVICE
U.S. DEPARTMENT OF COMMERCE
SPRINGFIELD, VA 22161

FOREWORD

The Ohio River Basin Energy Study (ORBES) is an interdisciplinary, multi-university inquiry into the environmental, economic, and social impacts of energy facility development in a major portion of the Ohio River Basin. The study was initiated by the U.S. Environmental Protection Agency in 1976 in response to a directive from the U.S. Senate Appropriations Committee.

Most of the ORBES research was conducted by a group of faculty members at the University of Illinois, Indiana University, the University of Kentucky, the University of Louisville, The Ohio State University, the University of Pittsburgh, Purdue University, and West Virginia University. In a few instances, the necessary expertise was found at other universities or independent research organizations.

This report is one of a series produced in support of the overall ORBES assessment. For a summary of the entire study, see Ohio River Basin Energy Study (ORBES): Main Report, by the ORBES Core Team, also being published by the U.S. Environmental Protection Agency.

DISCLAIMER

This report has been reviewed by the Office of Research and Development, U.S. Environmental Protection Agency, and approved for publication. Approval does not signify that the contents necessarily reflect the views and policies of the U.S. Environmental Protection Agency, nor does mention of trade names or commercial products constitute endorsement or recommendation for use.

ABSTRACT

This report evaluates major public health impacts of electric power generation and transmission associated with the nuclear fuel cycle and with coal use. Only existing technology is evaluated; for the nuclear cycle effects of future use of fuel reprocessing and long-term radioactive waste disposal are briefly considered. The health effects of concern are those leading to definable human disease and injury; subjective factors, effects of pollutants on visibility, or effects on agriculture and aquatic life forms are outside the scope of this report.

Health effects are scaled to numbers of persons and activities associated with a nominal 1000 Megawatt (electric) plant fueled by either option. Comparison of the total health effects to the general public from the nuclear and coal cycles gives: nuclear, 0.03-0.05 major health effects per 1000 MWe per year; coal, 0.7-3.7 per 1000 MWe per year. Thus for the general public the health risks from the coal cycle are about 50 times greater than for the nuclear cycle. Health effects to workers in the industry are currently quite high. For the nuclear cycle, 4.6-5.1 major health impacts per 1000 MWe per year; for coal, 6.5-10.9. The two-fold greater risk for the coal cycle is primarily due to high injury rates in coal miners.

There is no evidence that electrical transmission, common to both fuel cycles, contributes any health effects to the general public, except for episodes where broken power lines come in contact with people. For power line workers, the risk is estimated at 0.1 serious injury per 1000 MWe per year.

CONTENTS

Abstract..... iv

Tables..... vii

Acknowledgements..... viii

1. Introduction..... 1

2. Human Health Impacts from the Nuclear Fuel Cycle..... 2

Mining..... 2

Uranium Milling..... 8

Uranium Hexafluoride Production..... 11

Enrichment..... 13

Fuel Fabrication..... 15

Mixed Oxide (MOX) Fuel Fabrication..... 16

Nuclear Power Plants..... 16

Fuel Reprocessing..... 20

Waste Management..... 22

Transportation..... 23

Summary of Exposures..... 24

Health Effects from Nuclear Cycle..... 24

References..... 30

3. Human Health Impacts from the Coal Cycle..... 31

Coal Mining..... 32

Coal Processing..... 34

Coal Transportation..... 34

Coal-Fired Power Plants..... 36

Summary of Health Impacts from Coal Cycle..... 41

References..... 43

4. Power Transmission and Its Health Effects..... 44

References..... 49

TABLES

Number		Page
1.	Radioactive Materials Handled per Year to Supply 1000 MWe Plant	3
2.	1977 Uranium Mine Exposure to Radon Daughters (Company Records)	6
3.	Uranium Milling Airborne Releases of Radioactive Materials (Excluding Tailings)	7
4.	Radioactive Effluents from UF_6 Production (Curies) per 1000 MWe LWR Requirements/Yr	14
5.	Personnel Radiation Exposure for Light Water Reactors Licensed by U. S. Nuclear Regulatory Commission	19
6.	Effluents Discharged from Nuclear Fuel Services Reprocessing Plant	21
7A.	Exposures to General Public in Nuclear Fuel Cycle per 1000 MWe Power Generation/Yr	25
7B.	Exposures to Workers in Nuclear Fuel Cycles per 1000 MWe Generated/Yr	26
8.	Health Impacts of Nuclear Fuel Cycle per 1000 MWe-Yr of Power Production	29
9.	Circuit Miles of High Voltage Transmission Lines in U. S., 1970	45

ACKNOWLEDGMENTS

In the preparation of this report, I acknowledge the great assistance of my colleagues, Professors Maurice Shapiro, Sati Mazumdar, and Julian Andelman of the Graduate School of Public Health. Especially valuable have been the literature searches and draft reports prepared by my student assistants, Carol Godfrey, Dennis Cosmatos, Irene Goldberger, Stephan Lanes, and Eugene Takahashi. Finally, I acknowledge valuable discussions of the issues with a great number of professional experts in a wide range of topics in this large and rapidly evolving field.

SECTION 1

INTRODUCTION

In this report I have summarized my assessment of the potential human health problems associated with electric power generation and transmission. I deal here only with existing technologies for power production. Because oil has never been a significant fuel for electric power generation in the Ohio River Basin area, this report has been limited to the coal and nuclear options. Obviously introduction of new technology, such as breeder nuclear reactors or coal gasification, will modify the relative importance of some of the health impacts.

The report deals only with health effects that lead to definable human disease or injury, and does not attempt to quantitate subjective factors such as esthetics, nor does it deal with effects of pollutants on plant or animal life. Thus the possible role of acid rain in producing detrimental effects on aquatic or other biota is not included in this study.

With the limited resources available for this evaluation, it has been necessary to depend on a number of reviews of relevant subjects to obtain data in a concise form, but I have attempted to rely on more than one in each specific area of interest in order to provide a divergence of viewpoint. This final product is strictly based on my own perception of the scientific evidence and is not necessarily in accord with views of the authors of reports from which I have taken data.

In presenting the health impact results, I have attempted where possible to scale these to fuel use or other quantities required for operation of nominal 1000 Megawatt-electric plants of current design. Because control of future exposures of workers and the general public to hazardous conditions may significantly affect health risks over the period of interest to the ORBES study, some attention is paid to the possible impact of reduction in exposures to occupational or public groups on future projections of risks to the year 2000.

SECTION 2

HUMAN HEALTH IMPACTS FROM THE NUCLEAR FUEL CYCLE

We consider here health effects from "normal" operations of all aspects of the nuclear cycle, even though in the ORBES region only certain phases of the cycle are actually carried out. These latter include uranium enrichment and fuel fabrication plants, as well as power reactors. The fuel reprocessing and high level storage stages are not now operating commercially in the U. S., storage of fuel elements mainly being done at power plant sites. For this reason it is currently difficult to assess health effects from this phase of the fuel cycle. It appears from experience with previous commercial reprocessing plants, as well as those now operating in foreign countries, that reprocessing might contribute significant exposures to radiation, both for workers and for the general public in the vicinity of the plants. With regard to high level waste storage, it is presently not possible to assign any exposure to be expected from the current long-term disposal options.

Table 1 gives the throughput of radioactive materials in the steps of the nuclear cycle, scaled for a typical 1000 Mwe reactor (1).

MINING

The two major types of uranium recovery practiced in the U. S. are strip or pit mining and underground mining. Open pit mining is used when the ore body lies under relatively soft material at depths up to a few hundred feet; underground mining is employed when the ore body is at depths greater than 400 feet or when it lies under rock which requires a great deal of blasting to remove. Although there are considerably more underground mines operating today than open pit mines, the latter generally have higher production capacities and slightly more than half of the uranium mined in the U. S. is from open pit mines. The size of open pit and underground mines varies over a wide range. The average capacity of operating open pit mines in the U. S. is about 1.5×10^5MT of ore per year, and the average capacity of operating underground mines is about 2×10^4MT of ore annually. The ore requirements for a 1000 Mwe light water reactor (LWR) are about 10^5MT (Table 1) per year.

Sources of Radiation Exposure

1). Direct radiation from the ore. The principal exposure is to gamma radiation from decay of the uranium or thorium series of heavy elements. The extent of exposure will depend on the richness of the ore. For open pit mining the workers receive less radiation (about half) compared to underground

TABLE 1

RADIOACTIVE MATERIALS HANDLED PER YEAR TO SUPPLY 1000 MWE PLANT

Operation	Nature of Product	Throughput/yr Metric tons (MT)	Other Waste
Mining	Uranium ore	90,000	Radioactive rubble in mine
Milling	U_3O_8	182	Tailings of ∿ 9000 MT/yr
Conversion	UF_6	270	
Enrichment	Enriched UF_6	52	Depleted uranium tails
Fabrication	Power plant fuel	35*	
Reactor	Spent fuel	35*	Low-level waste 230 MT
Reprocessing	High level solid waste	7	Plutonium for recycle

* Metric tons of heavy metals.

miners because they are more distant from the ore and are not surrounded by it. In general, with typical uranium ore conditions, underground miners accumulate about 1 rad/yr from gamma ray exposure (2). Because underground exposures to radon daughters range from one to several working level months (WLM)*/yr, and the working level month gives rise to a dose to the bronchial epithelium of about 6 rem on the average (3). In terms of lung cancer in the contribution of the whole body gamma exposure to risk is minor. A risk of cancer at other organ sites may be contributed by the gamma radiation in underground miners, but should be insignificant for surface miners except under unusual circumstances.

2). Airborne particulates and gases. Uranium and its daughters are released to the atmosphere when the ore body is exposed and broken up during either underground or open pit mining operations. The airborne particulate radionuclides discharged from the mines are rapidly reduced by atmospheric dispersion and approach normal background levels at the site boundaries. U. S. Bureau of Mines measurements of radon concentrations in existing open pit mines revealed no significant alpha concentrations, but for very rich ore bodies some elevation of radon daughters can occur in the working area. A report by the Atomic Energy Commission also estimated that atmospheric concentrations of radon from open pit mining were minimal outside the work area; where scaled to 1000 MWe production requirement these emissions are less than 10% of the underground mines. For an underground mine with a capacity of 1.5×10^5MT of ore per year, the amount of radon gas emitted to the open air is estimated to be approximately 18 Ci per day (4), or about 4,500 curies/yr scaled to the production for a 1000 MWe LWR. This release rate is about 10 times greater than that due to the final tailings release, as discussed below. If, in fact, this amount of radon resulted in exposure to the general population of radon daughters at about 70% equilibrium, the bronchial dose would be on the order of 800 person-rem. Such an exposure would be by far the greatest in the entire fuel cycle.

The mines are generally remote from population centers, however, and there is the question of dispersion of radon (physical half-life 3.8 days, which governs the short half-life daughters) before it can affect people who may breathe the daughters (see discussion below for mill tailings). From current estimates it appears that nearly all of the exposure to radon daughters occurs locally within tens or hundreds of miles from release. In view of the low population density in mining areas, the exposure to the general population is likely to be much less than the projection cited above. Even if one assumes that an estimate of only 10% would be applicable, however, the contribution of dose to human populations from mines is still very significant, even if one ignores any contribution of the Po-210 daughter of long-lived Pb-210 resulting from radon decay. I use the estimate of 80 person-rem per nominal plant for lack of a better value.

---

*The "working level" is a concentration of short-lived radon daughters giving rise to 1.3×10^5Mev of alpha radiation per liter of air. The Working Level Month is a measure of cumulative exposure and is defined as exposure to one Working Level for 170 hours (hours worked/mo).

3). Solid and liquid waste. The solid waste from underground mining is estimated to be about equal to the volume of the processed ore. The gamma radioactivity associated with the solid waste is very small, but rubble present in mine areas can contribute to the radon present. The water pumped from both open pit and underground mines will contain dissolved and suspended uranium and its daughters. The radioactivity in the mine water is estimated at 1% of that in the ore, but it is removed from the water and returned to the ground by ion exchange during seepage through the soil. Therefore, the radioactivity reaching surface drainage is not expected to reach any significant levels.

Occupational Exposure

A major source of health effects associated with the mining of uranium is the occupational exposure to radiation incurred by the underground uranium miners. These miners are exposed to radon-222 and its daughter products, which are short-lived radioactive isotopes that emit alpha radiation and are potential cancer-causing agents when inhaled.

The Mining Enforcement and Safety Administration (MESA) increased its radiation exposure monitoring during 1977. Inspectors collected 1857 radon daughter samples at 153 uranium mines (compared to 1976 figures of 1180 samples at 142 mines). Average concentrations of radon daughters have consistently decreased in uranium mines, with the average falling from 0.71 WL in 1975, to 0.58 Wl in 1976, to 0.51 WL in 1977, but still well above the standard of 0.3 WL.

Table 2 presents data collected from uranium mining companies of radiation exposures to miners in 1977. The records also show a decrease in exposure from 1.07 WLM in 1975 to 0.99 WLM in 1976 to 0.91 WLM in 1977. A discrepancy exists, however, between federal sampling results and company records. When the federal data of concentration of radon daughters measured in WL is converted into exposure data measured in WLM, the average exposure is 5.68 WLM in 1975, 4.64 WLM in 1976, and 4.08 WLM in 1977.

A special radiation exposures and recordkeeping audit to investigate this discrepancy was initiated in 1975, and continued until its completion in 1977. Twenty underground uranium mines employing 1604 miners were visited by MESA auditing teams. The results from the investigation are also summarized in Table 2, which confirmed that uranium miner exposure was significantly greater than indicated by company records, with an average exposure of 4.64 WLM. According to this audit, only about 1/3 of the mines were actually in compliance with the exposure standard (3.6 WLM/yr).

Film badge surveys in open pit mines indicate that external whole-body radiation doses are generally less than 0.6 rem/yr. Radon-daughter concentrations are generally less than 0.3 WL, with the average concentration being 0.06 WL, or less than 10% of that found in underground mines.

TABLE 2

1977 URANIUM MINE EXPOSURE TO RADON DAUGHTERS
(COMPANY RECORDS)

Total employment	Average exposure	Miners having exposure in indicated intervals, percent				
		0-1 WLM	1-2 WLM	2-3 WLM	3-4 WLM	>4 WLM
5,315	.91 WLM	61.8	21.7	12.4	3.8	0.2

Radon daughter concentration in underground uranium mines 1976-1977 (MESA Audit)

Total number of samples	Average concentrations	Maximum concentrations	Number and percent of samples in designated range of annual exposure				
			0-4 WLM	4-8 WLM	8-12 WLM	12-24 WLM	>24 WLM
1,882	0.58 WL	24.0 WL	653 34.7%	473 25.1%	329 17.5%	291 15.5%	136 7.2%

TABLE 3

URANIUM MILLING
AIRBORNE RELEASES OF RADIOACTIVE MATERIALS
(EXCLUDING TAILINGS)

Radionuclide	Release Rate Ci/day	Site Boundary* Air Concentration pCi/m^3
Uranium	.0005	.079
Thorium 230	.00027	.043
Radium 226	.00013	.043
Radon 222	1.13	11.($\sim$ 0.1 WL)

*Distance to site boundary assumed to be 600 meters.

In assessing current and future exposure, the number of underground uranium miners is assumed to be 6000 at the present time, and the average occupational exposure per year is assumed to be 4 WLM.

It is difficult to take account of significant changes in these exposure levels in future operations if the ore continues to be worked in small veins, with 50% or more from underground operations. A realistic appraisal suggests that with better controls demanded, 2 WLM/yr could probably be readily achieved with better mine ventilation. If a more restrictive standard than this were required, it could probably be achieved in some mines only by restricting individual exposure times, thus increasing the number of miners and not changing the total cancers resulting. It should be noted that projections of mining activity and exposures are not greatly influenced by decisions concerning use of plutonium recycle in power production.

In calculating the maximal lifetime lung-cancer risk from occupational exposure, we assume the miner works underground 40 years, beginning at age 20. A lower average exposure would undoubtedly occur, but would be accompanied by a larger number of miners exposed.

URANIUM MILLING

The next stage in the uranium fuel cycle is the conversion of the mined ore containing about 0.2% U_3O_8 into a compound containing 80 to 83% U_3O_8 (yellowcake). This process takes place at uranium mills, which are located near the mines in order to minimize ore haulage distances. As of January 1, 1976, there were 17 uranium mills in operation in the U. S. The majority of the mills are conventional acid leach or alkaline leach mills, but several obtain the uranium by solution mining or from phosphoric acid treatment.

Although the uranium extraction process may vary between mills due to the chemical composition of the ores, steps common to all mills processing conventionally mined ores include crushing, grinding, chemical leaching and recovery of the uranium from the leach solutions. The problem of radiological controls is present throughout this entire process, primarily because uranium and uranium daughter products are released to the atmosphere in the form of dust and radon gas (Rn-222).

After the uranium is extracted from the ore, more than 99% of the ore material becomes the mill wastes or tailings, a slurry of sandlike material in waste solutions. The tailings are pumped to nearby sites where the solids settle out and soon accumulate to form a tailings pile.

Radiological Effluents

The radioactivity associated with uranium mill effluents are emitted from natural uranium and its daughter products (thorium-230 and radium-226) present in the ore. The milling process removes and concentrates the bulk of the

natural uranium, while the majority of the radioactive daughter products remains in the uranium-depleted solid wastes which are piped to the tailings retention system.

A). Airborne. Radioactive airborne effluents consisting of uranium and its daughter products are released primarily from the ore crushing and grinding ventilation system, and from the yellowcake drying operations in the form of dust. Additional particulates and radon gas are also released from the tailing piles near the mill, which will be discussed in a later section. Table 3 shows the airborne effluents which are estimated to be released from milling operations.

It is estimated that each mill is associated with about 5000 MWe generating capacity, therefore the mill radon output (excluding tailings) per 1000 MWe nominal plant is about 85 Ci/year, very much lower than for the mine or tailings releases.

B). Liquid. The liquid effluents from a mill (acid-leach) consists of waste solutions from the leaching, grinding, extraction and washing operations of the mill. This liquid is discharged with the solids into the tailing pond, which is designed to prevent contamination of ground or surface waters. About 4300 m^3 of waste solutions are generated per day, and the amounts of heavy radionuclides released in this waste water would be about 2 m Ci of natural uranium, 1 m Ci of Ra-226 and 50 m Ci of Th-230 per day. The tailings are composed mainly of sandstone and clay particles, and constitute about 1/3 of the weight of the slurry.

C). Releases from the mill tailing piles. The mill tailing piles are the major source of radioactive materials released to the environment from mill sites. The solid waste tailings contain about 75-85% of the radioactive materials originally in the ore. While stored in the tailings retention pond, low-level activity continues to decay naturally, resulting in further discharge of radioactive effluents to the atmosphere. Radon-222 generated by the decay of Ra-226 parent and Th-230 grandparent represents a significant localized source of radon from annual fuel requirements for a 1000 MWe plant, although not all of it will reach the atmosphere.

Studies of concentrations in air near mill tailing piles have indicated that at distances beyond 0.5 mile in the prevailing wind direction, radon concentrations are not significantly higher than background levels in these areas. The rate of radon gas escape to the atmosphere from the tailings site of a 1500 MT yellowcake mill after 10 years of operation is estimated to be about 3000 Ci/yr. Scaled to a nominal plant this is about 400 Ci/yr per 1000 MWe. The rate may vary with location, however, due to the effects of weather conditions such as wind or precipitation. A pile saturated with water may have its radon release rate decreased by a factor of 25 because of decay during diffusion to the surface. Unlike the radon releases from underground uranium mines, which for all practical purposes end when the mine is shut down and sealed off, releases from mill tailings continue after the mill has been closed. This is because one source of the radon gas, thorium-230, has a very long radioactive half-life (8×10^4 years).

The tailing piles also release radioactive materials such as waterborne radionuclides leached out by precipitation, surface runoff and waste solutions. Although the movement of radioactive materials through seepage can be expected to be very slow, seepage surveillance is required to prevent unacceptable contamination of local water sources. Leaching into human water supplies can constitute a significant human exposure route.

Occupational Exposures

Film badge surveys conducted in uranium mills indicate that external whole body radiation doses to employees are generally less than 600 mrem per year. Although there are only a limited number of surveys measuring internal exposure received by mill workers, the indication is that significant radon daughter concentrations are usually not found. There exist a few exceptions in which the concentrations approach those experienced in underground mines, but with improved ventilation it is assumed that future exposure to radon daughters in uranium mills should be low. Nevertheless, projections to date indicate that total lung exposures to radon daughters and other alpha-emitters in milling operations could be similar to that for mining.

Population Exposures

EPA has estimated the population weighted exposure from radon daughters arising from uranium mill tailings using two different models for atmospheric transport (5). The simplest model, which treats radon diffusion on the basis of gross meteorology, yields an annual exposure of 6.5×10^{-4} person WL per curie/yr released. A more detailed meteorological model developed for EPA by the National Oceanic and Atmospheric Administration (NOAA) has been used to calculate the exposure due to radon daughters from four release sites in the western U. S. These exposures range from 4.2×10^{-4} to 7.6×10^{-4} person-WL per curie per year released, yielding an average of 5.6×10^{-4}. A population exposure due to radon daughters of 6×10^{-4} person-WL per curie released per year is used in this analysis. It is based on the assumption that exposure from the emitted radon and its daughters occur at seven tenths of the equilibrium concentration. This average applies to the entire U. S. population, although it appears that the local population is primarily affected. In 1974, EPA estimated the annual release from twenty-two known inactive uranium mill tailings piles as about 80,000 Ci per year. The population exposure from this release would be estimated as

$$6 \times 10^{-4} \frac{\text{person-WL}}{\text{Ci/yr}} \times 8 \times 10^{4} \text{ Ci/yr} = 48 \text{ person-WL}$$

The individual exposures to persons working or residing near the point of release are of course substantially larger than the national average. Because the local population density is markedly different near various tailings piles, the regional impact from each should be considered on a site specific basis. When it is arbitrarily assumed that tailings piles are located in a low-population area, 7.5 persons per square mile, the population exposure to a uniformly distributed local population is about 1×10^{-4} person-WL per curie/yr released, significantly lower than estimated above. However, a preliminary EPA analysis of the actual population distribution around tailings piles located at Salt Lake City, Utah, and Grand Junction, Colorado, indicates that the regional population exposures from these sources account for almost all the exposure.

In those cases where the local population density is high and the tailings pile large, the magnitude of regional health impact projections approach those for the national total from all inactive tailings piles. For example, the Vitro pile in Salt Lake City, which produces a large fraction (16%) of the total radon emissions in the U. S. from inactive piles, results in an exposure of about 100 person-working levels to the 800,000 persons who reside locally. For Grand Junction, Colorado, the estimated exposure to the regional population is about 40 person-working levels per year.

To convert person-working levels to person-rem, we assume continual exposure to radon daughters to calculate WLM, and a value of 6 rem/WLM (3). On this basis 1 person-working level = 300 person-rem dose to the bronchial epithelium per year.

URANIUM HEXAFLUORIDE PRODUCTION

The yellowcake extracted from the ore in the milling process is shipped to conversion plants, where the U_3O_8 concentrate is converted into the volatile compound uranium hexafluoride (UF_6). The UF_6 is then used as feed to an enrichment facility. The total domestic commercial industry produces about 17,300 MT of uranium per year as UF_6, with a typical conversion plant having a capacity of 7000 MT of UF_6 annually. This capacity is sufficient to produce 24,000 MWe-yr of electric energy.

There are two different industrial processes involved in the production of UF_6. In the dry hydrofluor process, the uranium concentrate is passed through successfive reduction, hydrofluorination and fluorination steps

followed by fractional distillation of the crude uranium hexafluoride to obtain a pure product. The second method begins with a wet chemical solvent extraction step to produce a highly purified uranium oxide feed prior to the reduction, hydrofluorination and fluorination steps. In this process, a flame reactor is used instead of a fluidized bed unit to carry out the fluorination step. Approximately equal quantities of UF_6 feed are produced by each method, although waste effluents differ somewhat between the two methods.

Radiological Effluents

In the dry hydrofluor process the radioactivity released is primarily in the gaseous and solid states, while the wet solvent process discharges radioactivity into liquid streams.

1). Gaseous effluents. A minimal amount of radioactive effluents are released into the atmosphere from exhaust air as dust. The quantity of natural uranium emitted in this fashion from all processes associated with the wet procedure is approximately 100 mCi per year. The calculated concentration of natural uranium in the air at the site boundary is about 700 pCi/m^3. The total industry airborne emissions are estimated to be about 8 Ci for the period 1975-2000.

2). Liquid effluents. Radioactivity in liquid effluents, originating primarily from natural uranium, has averaged less than $2\mu Ci/m^3$. The radionuclide concentration in the water retention pond of the wet solvent process would be about 1 mCi/m^3 of Ra-226 and lesser amounts of Th-230 and natural uranium. The radioactivity in a neutralized pond is usually found in the settled sludge, with only about 2% of its total activity present in the liquids. After evaporation, the remaining sludge is expected to be disposed of at licensed burial grounds or reprocessed at a mill to recover the uranium, with transfer of the waste material to the mill tailings pond.

The wet process UF_6 facility has a river adjacent to it, whose flow has been measured monthly. The flow would be expected to provide enough dilution of any radionuclides in it to reduce their concentration to below maximum permissable levels.

Estimated waterborne radiological emissions for all heavy elements (mainly U-238, U-234, and Th-234) from the projected UF_6 industry range from 340 Ci to about 451 Ci, depending on whether plutonium recycle is used or not.

3) Solid effluents. The primary source of solid radioactive wastes is the ash residue from the fluidized bed fluorinators formed during the dry

hydrofluor process. The residue is packaged and consigned for burial at a licensed commercial disposal site. The principal constituent is Th-230 (18 Ci/yr) with about 3 Ci/yr of uranium isotopes in equilibrium with their daughters, and about 2 Ci/yr of Ra-226. Much smaller amounts of solid ash wastes are produced in the flame fluorination stage of the UF_6 wet solvent process. The total quantity of radioactive solids estimated to be emitted by the UF_6 industry over the period from 1975-2000 ranges from 21,000 Ci to 28,000 Ci.

Table 4 summarizes the radiological effluents from hexafluoride production of both types, normalized to a model 1000 MWe-yr LWR annual fuel requirement.

Occupational Exposure

Employees of the UF_6 portion of the uranium fuel cycle are exposed to external beta and gamma radiation, as well as to natural uranium and uranium daughters in the atmosphere. Domestic UF_6 plant operations have conducted plant measurement surveys, and their results indicate that external exposure to beta and gamma radiation averages about 0.5 rem/yr, but most of this is from the beta component and thus does not apply to the whole body problem.

Internal body doses are the result of inhalation of air containing radionuclides in addition to penetrating external gamma radiation. An average whole body dose commitment per individual worker is expected to be about 0.2- rems/yr with critical organ dose commitments of about 1.5 rems to the lung and about 2.2 rems to the bone. These figures can be used to compute an integrated whole body dose commitment for the entire industry population over the period 1975-2000 of 3500 to 4400 person-rems.

ENRICHMENT

Enrichment of uranium is presently being carried out at 3 large government-owned and contracted gaseous diffusion plants, located at Oak Ridge, Tennessee, Paducah, Kentucky and Portsmouth, Ohio. The total capacity of the 3 plants have an output which is sufficient to produce about 90,000 MWe-yr of electrical energy from LWRs. The federal government is currently planning to increase the capacities of the 3 plants by a factor of 2.5 to meet projected domestic and foreign demands. Conversion to the gas centrifuge method is being considered.

Radiological Effluents

Small amounts of uranium are released from the enrichment plants during normal operations in both gaseous and liquid forms. The bulk of the radioactive effluents are discharged to holding ponds as a liquid and then dispersed to streams, while a small portion of the effluents are released to the atmosphere. Releases from uranium enrichment normalized to a 1000 MWe-yr model LWR annual fuel requirement is about 0.002 Ci/yr of uranium in gaseous forms and 0.01 Ci/yr of uranium in liquids.

TABLE 4

RADIOACTIVE EFFLUENTS FROM UF_6 PRODUCTION (CURIES)
PER 1000 MWE LWR REQUIREMENT/YR.

Gases	
Uranium	0.003
Liquids	
Ra-226	0.0034
Th-230	0.0015
Uranium	0.044
Solids (buried)	
Other than high-level	0.86

Occupational Exposure

Personnel employed in the uranium enrichment industry will be exposed to small quantities of beta and gamma radiation, as well as radioactivity from airborne particles. Annual monitoring of personnel at the Oak Ridge Gaseous Diffusion Plant was made over the five-year period, 1970-1974. Exposure to gamma radiation has remained at about 50 mrem/yr on the average, but the beta doses have declined sharply from 1970 to 1974 and now add little to the effective dose. Internal exposure can result from inhalation of radionuclides in the air. The level of the radionuclides does not exceed 5.5 pCi/m^3, which will produce an annual critical organ dose commitment of about 0.7 rems/individual.

Predictions for the year 2000 indicate that the occupational doses will not change significantly for the gaseous diffusion operations. For plants using the centrifuge procedure, however, the whole body occupational dose commitments are expected to increase because of the increased number of employees at these facilities.

FUEL FABRICATION

The enriched uranium hexafluoride is used as feed material in the fabrication of fuel for LWRs. It is converted to UO_2 powder, which is formed into pellets, sintered and then loaded and sealed into zircaloy or stainless steel tubes. The process that is currently being used to produce the fuel is a wet ammonium diuranate (ADU) process. The present fuel fabrication industry consists of 9 commercial operations, each performing either a specific step of the fabrication process, or the entire process.

Radiological Effluents

Very small quantities of airborne radionuclides are emitted from fabrication operations. The annual atmospheric releases for a plant with an annual throughput of 1000 MT or uranium are estimated to be about 0.008 Ci/yr. About one-third of the uranium losses are liquid and are discharged to a waste lagoon. Liquid releases include uranium and thorium-234 (which grows back after being separated from the uranium products at the milling plant). The solid wastes are made up of CaF_2 containing small amounts of uranium, and account for almost two-thirds of the uranium losses. These solid effluents are normally buried within the plant site, or may be sent to licensed commercial burial grounds. The radiological effluents from a fabrication plant normalized to a model 1000 MWe-yr LWR annual fuel requirement are about 0.002 Ci/yr of uranium in gas form, 0.013 Ci/yr of uranium and thorium-234 in liquids, and 0.23 Ci of uranium in solids.

The estimated radioactive emissions from the fuel fabrication industry over the period 1975-2000 are predicted at 1.3 curies from airborne effluents and 3.0 curies from liquid effluents if no recycle is implemented, and a slight reduction if both uranium and plutonium are recycled.

Occupational Exposure

Personnel employed by the fuel fabrication industry are exposed to external radiation, as well as to internal radiation from airborne radionuclides. These exposures are estimated to produce an annual whole body dose commitment of about 0.20 rem per individual, in addition to a critical organ dose commitment of about 10 rems to the lung.

MIXED OXIDE (MOX) FUEL FABRICATION

If plutonium recycle is incorporated into the uranium fuel cycle, then fabrication of mixed oxide fuel from uranium and recovered plutonium will be required. It is estimated that 8 model plants employing 24,000 persons would be needed for this to occur by the year 2000. It is assumed that plutonium recycle will not begin before the year 1981.

Radiological Effluents

Estimates of radiological effluents emitted from a MOX fabrication industry are about 0.034 Ci of uranium, plutonium and americum in the gas phase, and about 0.007 Ci of the same mixture as liquid waste.

Occupational Exposure

Employees of a MOX fuel fabrication plant may receive an average annual whole body dose commitment of about 1.2 rems from external radiation and 0.2 rems from internal radiation.

NUCLEAR POWER PLANTS

Within the reactor radioactive materials are generated as a result of the fissioning of uranium, and by neutron absorption in the coolant and structural materials. The products of uranium fission include a large number of elements in both their stable and radioactive forms. In the operation of nuclear reactors, a small amount of these created radioactive materials is expelled to the atmosphere or to nearby waters, but the majority are contained within fuel rods. After a period of operation, the spent fuel rods must be removed from the reactor, stored for a cooling period and then shipped to a fuel reprocessing plant. The amount of radioactive effluents released to the environment at the reactor site depends on several factors, such as the type and size of the reactor, reactor design, percent of fuel failure, and the control measures used.

Almost all of the current nuclear power plants in operation in the U. S. today employ light water reactors to generate power. There are 2 types of LWRs--the Boiling Water Reactor (BWR) which allows the water in the reactor to boil, and the Pressurized Water Reactor (PWR) where the water inside the reactor is prevented from boiling by pressures of about 150 times atmospheric pressure. These differences in processes result in different types and quantities of radioactive materials being emitted from PWRs and BWRs.

Radiological Effluents

Radionuclides may enter the environment through either the gaseous or liquid effluent streams. The airborne effluents consist of fission noble gases (krypton and xenon isotopes), activation gases (Ar-41, C-14, N-16, S-35), tritium, radioactive halogens and particulates. In the liquid effluents are found tritium, fission products and activated corrosion products.

A). Fission noble gases. Several isotopes of the noble gases Kr and Xe are created by fission in the nuclear fuel. The amount and composition of Kr and Xe escaping to the coolant and eventually to the environment depends on the rate of their escape and the hold-up time prior to release. In PWRs the gas volume buildup in the coolant (water flowing through the core) is relatively slow, and therefore gases can be held up for a relatively long time (more than 30 days) before being released. A 30-day delay would reduce all the short-lived radionuclides to an insignificant level and only Kr-85 and Xe-133 would be released in any measurable quantities. The gas production rate in the BWR coolant, however, requires continuous removal of non-condensable gases and, because of its large volume, the time between gas removal and release must be shorter. The current hold-up time for krypton from BWRs is about one day. In addition to Kr-85, Xe-133 and Xe-135 comprise an important part of the noble gas effluents from BWRs. The total emission of noble gases from PWRs during 1976 varied from a few up to 15,000 Ci/1000 MWe/yr, whereas from BWRs the range was from 30,000 to more than 1,000,000 Ci/1000 MWe/yr.

B). Tritium. Tritium is produced in both BWRs and PWRs from tertiary fission, but most is produced as an activation product in the primary coolant. Approximately 90% of the tritium is released as a liquid effluent and the remainder in the form of a gaseous water vapor. About 1000 Ci/1000 MWe-yr was released from PWRs and 100 Ci/1000 MWe-yr from BWRs.

C). Radioiodine. Several isotopes of iodine are produced in reactors by fission and by decay of other fission products. I-131 with a half-life of 8 days is the isotope which is of main concern from an environmental standpoint. It is released as both a liquid and gaseous effluent. The measured releases of this radioactive halogen vary from several millicuries to up to 100 curies per 1000 MWe-yr of power production.

D). Particulates in airborne effluents. Particulates can arise either indirectly or as decay products of fission noble gases. Almost all of the particulates are prevented from escaping to the atmosphere by high efficiency particulate air filters. Only the smallest particles can escape, which results in very low levels of particulate activity. It is estimated that in 1974 from PWRs, there was 0.5 Ci per 1000 MWe/yr emitted, and 1.1 Ci per 1000 MWe/yr from BWRs in the U. S.

E). Radionuclides in liquid effluents. Tritium and I-131 comprise the major portion of the liquid effluents from reactors, but there are other radionuclides which can be found in significant quantities. These include a number of fission and activation corrosion products, such as cesium-137,

cesium-134, cobalt-58, cobalt-60, chromium-51, magnesium-54, molybdenum-99, and iron-55.

The U. S. Nuclear Regulatory Commission annually presents measured data on radioactive materials in effluents from licensed commercial LWR power plants. EPA has studied time trends of releases, which indicate the fluctuations which have occurred in the release of radioactive effluents over a 5-year time span from 1970 to 1975. There is also great variability in releases from one reactor to another of the same type.

Occupational Radiation Exposure at Power Reactors

Table 5 is a summary of the number of annual whole body exposures to workers at LWRs for the years 1969 through 1977. The total number of individuals monitored for radiation exposure has increased steadily, as has the number of personnel receiving measureable exposure. The percent of individuals with measurable dose receiving exposures exceeding 5 rems has remained about 1% over recent years.

An average of 669 persons per reactor received measurable exposure at commercial LWRs in 1976, a number which represents a progressive increase during the past several years. Since 1972 the average exposure to workers has remained about 0.7-0.8 rem/yr. The number of persons receiving exposures in excess of the short-term limits established by 10 CFR 20 at commercial reactors is also reported for the years 1971 through 1977 in Table 5. It is apparent from Table 5 that both in terms of short-term overexposures (most of which result in partial body exposure from inhaled radionuclides, generally of a few rads), or in terms of average annual exposures, the current standards are frequently exceeded.

The data shown in Table 5 are for all reactors operating in each year. Analysis by specific reactor shows that while the average dose per exposed worker for all reactors is about 0.75 rem/yr, for individual reactors the averages show a wide range. In 1976, the range of average exposures for BWRs was 0.4 to 2.2 rems, while for PWRs the range was 0.2 to 1.7 rems. In part this range reflects the number of years the reactor has been in operation, the older plants having higher exposures to the workers, but the large range shows that control of worker exposure is much less effective in some plants than in others.

Within a particular plant there is also variability in radiation exposure by the type of work carried out. Evidence indicates that maintenance workers and the health physicists responsible for radiation protection have higher individual doses than operating personnel in the reactors. The significance of this variation is that simply giving an average annual exposure cannot indicate the potential risk to individual workers in a facility, although it is a useful datum for analysis of total societal cost of nuclear operation. The consistency of the pattern of exposure shows that definition of individual risk by job category and by reactor can be done, but there remains the question of the appropriate level to assume, especially when projecting to the year 2000.

TABLE 5

PERSONNEL RADIATION EXPOSURE FOR LIGHT WATER REACTORS LICENSED BY U.S. NUCLEAR REGULATORY COMMISSION

SUMMARY OF ANNUAL WHOLE BODY EXPOSURE BY INCREMENT

YEAR	TOTAL MONITORED	NUMBER OF INDIVIDUALS EXPOSURE INCREMENT - REM												NUMBER OF OVER-EXPOSURES
		NOT MEASURABLE	0-1	1-2	2-3	3-4	4-5	5-6	6-7	7-8	8-9	9-10	>10	
1969	2854	2607			144	70	26	5	2	0	0	0	0	
1970	7518	6953			184	175	92	102	11	1	0	0	0	
1971	10269	9660			328	146	107	17	11	0	0	0	0	2
1972	15730	14783			536	205	114	47	23	10	6	6	0	16
1973	35918	20717	10249	2449	1585	432	237	117	71	38	16	7	0	19
1974	38379	20240	13455	2491	1375	470	226	86	30	6	0	0	0	43
1975	45659	20188	18277	3892	1903	707	426	169	60	24	12	0	1	14
1976	61151	25704	26636	4880	2354	789	487	188	70	26	11	5	1	20
1977	70904	27671	33252	6174	2838	1130	569	141	66	36	21	6	0	27

SOURCES: 1969-1976: NUREG-0323 Occupational Radiation Exposure at Light Water Cooled Power Reactors 1976
1977: NUREG-0463 "Occupational Radiation Exposure" Tenth Annual Report 1977

FUEL REPROCESSING

Spent fuel from uranium-fueled LWRs can be reprocessed to recover usable fissionable material (both uranium and plutonium). The fuel elements removed from the reactor still contain about one-quarter of the original U-235, in addition to Pu-239 produced from the absorption of a neutron by U-238. If plutonium recycle is implemented, a portion of the original plutonium content will also remain in the spent fuel. The recovered uranium is converted to UF_6, shipped to enrichment plants and eventually recycled to nuclear reactors. The recovered plutonium can be converted to PuO_2 and then mixed with UO_2 for recycle in reactors. The reprocessing operation involves the separation of the fission products from the uranium and plutonium. The fissionable uranium and plutonium are recovered as purified nitrate solutions. Radioactive wastes must be converted into forms suitable for transfer to federal waste repositories, which will be the ultimate confinement location for various types of radioactive wastes.

Three commercial nuclear fuel separation facilities have been built in the U. S., none of which are in operation at the present time. Reprocessing and recycling have been controversial subjects because of the potential health problems which can result from the widespread use of plutonium, and because plutonium can be used in the production of nuclear explosives.

Radiological Effluents

After the fuel rods with the spent fuel are removed from the reactor, they are stored for about 150 days to allow significant reduction of the radioactivity of the shorter half-life radionuclides. Despite this reduction in radioactivity, however, large quantities of natural and reactor-generated radioactive materials are still found in fuel rods at reprocessing. A significant portion of the radioactive effluents from the entire fuel cycle is encountered in the reprocessing stage.

The Nuclear Fuel Services Reprocessing Plant (NFS) was in operation from 1966 to 1972, and was then shut down for alterations and expansion. The plant released to the atmosphere all the Kr-85 and C-14, about 60% of the H-3, and very small quantities of radioiodine and other fission products present in the spent fuel. Additional tritium and other radioactive materials were released to streams as liquid effluents. The following data in Table 6 show the effluents discharged from the NFS plant, normalized to a 1000 MWe-yr of uranium processed.

If plutonium is recycled in LWRs over the period from 1975 to 2000, the amount of MOX reprocessed is expected to total about 11% of the total fuel reprocessed, mostly to be processed in plants not yet designed. The recycling of plutonium will not affect the amount of fuel reprocessed, but will primarily change the mixture of radioactive effluents.

Determination of the population or occupational exposures from fuel reprocessing is difficult at this time. From the nature of the isotopes released at NFS (Table 6) exposures to the general population would be

TABLE 6

EFFLUENTS DISCHARGED FROM NUCLEAR FUEL SERVICES
REPROCESSING PLANT

Radionuclide	Ci/1000 Mwe-yr
Gases	
Krypton-85	340,000
Tritium (H^3)	220,000
Carbon-14	10
Iodine-129	0.003
Iodine-131	0.06
Other fission products	0.9
Actinides	0.004
Liquid	
Tritium (H^3)	4,000
Ruthenium-106	4

largely to krypton-85, tritium and transuranic actinides, and thus would affect mainly people living near the plants. Estimates of occupational exposures are generally low, but experience with the reprocessing plant in England, which has been operating commercially for more than a decade, shows average occupational exposures of about 1.3 rads/yr.

WASTE MANAGEMENT

The disposal of radioactive wastes is one of the most controversial aspects of the uranium fuel cycle. The major problem is presented by the highly radioactive fission products created in the nuclear reactors, as well as the plutonium and other heavy elements produced there.

The Nuclear Regulatory Commission has grouped the wastes from the nuclear fuel cycle into six major categories. Four of these arise from either uranium-only recycle or both uranium and plutonium recycle: high-level wastes, transuranium-contaminated wastes, non-transuranium-contaminated wastes (usually called low-level wastes), and contaminated facilities and large equipment. In addition, the uranium fuel cycles give rise to two other classes or wastes: spent fuel from the no-recycle option and unused plutonium from the uranium-recycle option.

High-level wastes (HLW) are produced at fuel reprocessing plants, and contain most of the highly radioactive fission products separated from fissile material. These wastes are perhaps the most difficult to handle in terms of containment because of the need for shielding and heat dissipation. According to Appendix F of 10 CFR 50, HLW must be converted to solids within 5 years after being generated, which then must be transferred to a federal repository for storage within an additional 5 years. The fission products that are found in HLW include the transplutonium elements and about 0.5% of the uranium and plutonium that were originally in the spent fuels.

Transuranic wastes are determined by their concentration of transuranic elements (i.e., a minimum of 10 nCi/g being emitted from elements with atomic numbers larger than 92). Transuranic wastes are generated primarily after fuel reprocessing operations. Waste materials included in this category are solidified liquids, cladding hulls and other fuel hardware, and general trash.

Non-transuranic low-level wastes are defined as any radioactive waste material which contains less than the minimum amount of transuranium elements (as defined in the previous paragraph). These materials are contaminated with fission products with relatively short half-lives, the maximum being around 30 years.

Contaminated equipment and large facilities are taken out of service by a procedure termed decommissioning, which varies depending on the type of facility. Some of the actions involved in decommissioning include equipment removal, removal of contaminated concrete, chemical scrubbing and removal of cell liners.

Wastes containing radioactive isotopes are produced at each segment of the uranium fuel cycle. Much of the waste is low-level and can be dealt with by storage in licensed repositories, but the principal problem is potential exposure to the general public from solidified high-level wastes and the transuranics. The bulk volume of this waste is relatively low, about 10 m^3 of spent fuel per 1000 MWe-yr for current reactors, and less for solidified fission products from reprocessing. Long-term storage in a location from which negligible activity can reach human contact is required.

At the present time, the plan most carefully researched for long-term disposal of HLW calls for placement in geologically stable salt beds. The Department of Energy is in the process of selecting a suitable site for this disposal. If one assumes that a repository for HLW or spent fuel can be built which will adequately cope with the long-lived radionuclides generated in the production of nuclear fuel, the environmental impact of nuclear waste management should be relatively small, but at present this problem is unresolved.

TRANSPORTATION

Throughout the nuclear cycle, transportation of potentially hazardous radioactive materials is required, and for this reason the total number of miles of transport is higher than for coal. In general, however, the volume of material transported is much less than for the coal cycle, but the potential hazard to the general public or the workers is greater than for coal, especially if the containment of these materials is breached. In addition, there is the potential for injury due to accidents even if the containment of activity is maintained.

The most significant source of exposure is from spent fuel or reprocessed fission products after the fuel is used in energy production. The risks from spent fuel will predominate in the short run, but in the long run transportation of the reprocessed waste will probably lead to greater population risks than for spent fuel. Transportation of these highly radioactive sources is carried out in lead-shielded casks designed to withstand significant impact. Shipment is primarily by truck, but some by rail. To the present, we can only speculate on the health impact on the general population of transportation of nuclear materials, except from gamma ray exposures in transit, which are less than 1 mrem per person, or negligible in relation to natural background. An individual who stood next to a cask for several minutes could receive about 1 mrem/2 min. Exposures of the general population from accidental breaching of the containment casks could be much higher or not, depending on where the accident occurred, but it is difficult to forecast these kinds of exposures.

Occupational radiation exposures are relatively low. Estimates of exposure for the truck drivers are about 15 mrem for a 500-mile haul, including time spent working adjacent to the cask. Spent fuel haulage is estimated to require about 20 shipments per year per 1000 MWe, or a total of 10,000 miles per year per nominal plant. Thus at the present rate, exposures of drivers should be well within acceptable limits, unless relatively few drivers handle the entire nuclear industry. If plutonium recycle is undertaken, some additional exposure would occur, particularly from transport of the solidified high-level

waste to a final repository. Finally, risks to the drivers from road accidents exist, but they are likely to be much lower per shipment than for coal shipments because the progress of the shipment will be very closely supervised.

The above discussion has assumed that shipments of ore or unirradiated fuel contribute negligible population or worker exposures. Low-level waste shipments may contribute some occupational exposures because of the larger volume and greater number of shipments required per 1000 MWe/yr.

An accident occurred in Colorado in 1977 in which several tons of yellowcake (U_3O_8) were spilled. This incident did not produce a significant general exposure to radiation, but it did emphasize the importance of plans for clean-up operations from a major spill, especially if the material had a much higher radiation risk than yellowcake.

SUMMARY OF EXPOSURES

From the preceding discussion we may summarize the radiation exposures in the nuclear fuel cycle. The following data are reasonably consistent with the recent EPA estimates of airborne exposure (4). In considering health impacts, obviously hazards not related to radiation must also be considered. In Table 7, these exposure factors are summarized for the nuclear cycle. Table 7A gives general population exposures and Table 7B gives occupational exposures.

These exposures are on a yearly basis. For some of the long-lived radionuclides the integrated total dose, or dose commitment, is used to express exposure. This is a necessary approach to an individual when the exposure leads to a continuous integral dose from nuclides present in the body over many years, and it has been used here. In the sense, however, that annual releases of radiation exposure sources sum over time to give a very long exposure, as in the case of uranium mill tailings, it is evident that single individuals will not experience the entire cumulative effect which may last for centuries. For this reason, I have elected to emphasize the dose relevant to individuals, but this approach does not give stress to the very long-range impact of the nuclear cycle. The situation is analogous to the long-range effects of strip-mining for coal, which may alter the environment permanently, conceivably with health impacts a century hence. If there were significant gonadal dose from the mine tailings, the long-range genetic effects of radiation could be important, but there is no evidence that this is the case.

HEALTH EFFECTS FROM NUCLEAR CYCLE

In this section I have applied risk estimates based on the linear no-threshold dose-response relationship, derived from the draft 1979 BEIR report (3). These risk coefficients are applied to the exposure data in Tables 7A & 7B which are normalized to a nominal plant.

TABLE 7A

EXPOSURES TO GENERAL PUBLIC IN NUCLEAR FUEL CYCLE
PER 1000 MWE POWER GENERATION/YR.

Location	Total (person-rem)	Individual Maximum (rem/yr.)
Mining	80 (lung)	very low
Milling (tailings)	70 (lung)	0.035 (lung)
Conversion	2 (lung)	0.001 (lung)
Enrichment	Minimal (<1)	0.006 (lung)
Fabrication	Minimal (<1)	0.005 (lung)
Power plants	20	0.01 (thyroid)
Reprocessing	200 (est)	0.1 (lung)
Waste management	10 (est)	unknown
Transportation	3	0.001 (whole body)

TABLE 7B

EXPOSURES TO WORKERS IN NUCLEAR FUEL CYCLE
PER 1000 MWE GENERATED/YEAR

Location	No. of workers	Total exposure person-rem (whole body)	Individual average of exposed group (rem/yr.)	Other hazards
Mining				
Underground	∿200	200	25 (lung) 1 (whole body)	Accidents, Silicosis
Surface	∿100	50	0.5 (whole body)	Accidents
Milling	∿30	5	10 (lung) 0.2 (whole body)	
Conversion	∿80	16	3 (lung) 0.2 (whole body)	
Enrichment	∿500	25	0.05 (whole body)	
Fuel fabrication	∿500	100	10 (lung) 0.2 (whole body)	
Power plants	800	800	1 (whole body)	Accidents
Reprocessing		unknown	∿1 (whole body)	
Waste management		unknown		
Transportation	∿5	1	0.2 (whole body)	Accidents
Total	∿ 2000	1200		

General Population

For these exposures (Table 7A), health effects are mainly from effluents of mining, mill tailings, fuel reprocessing and power plants. A single-dose exposure lifetime-risk model is appropriate. This model has been applied to a population of all ages and both sexes, by application of life-table methods. The principal health effect of radiation is cancer production. It is not possible to quantitate in health terms genetic effects. Thus no attempt is made to do so, but it should be borne in mind that estimates for cancer alone do not fully indicate the health implications of radiation exposure. In deriving cancer incidence, projection of lifetime risk has a range of uncertainty because the appropriate model for lifetime projection is unknown at this time.

Because fuel reprocessing is not currently under way, this phase and waste disposal are considered separately. For those stages of the cycle currently operating, most exposure is from mining and milling effluents, although there is considerable uncertainty about the contribution of mining. The total incident life-time cancers expected from a single year of exposure would be about 0.028-0.048 cases per year. For fuel reprocessing, the excess cancer incidence from a single year's exposure would be about 0.08-0.25 cases per year, probably in a smaller exposed population.

Occupational Health Effects

Occupational health effects can be more definitely specified than for the general population. In this case the assumption is made that workers begin at age 20 and retire at age 65, and most of the workers are likely to be male. In this case no estimate is made for the reprocessing workers, although it is likely that they will contribute significantly in the future. In addition, for mining we assume 2/3 of the miners are underground, and the lung cancer risks in fuel fabrication are comparable per rem to that in the mining and milling operations, an assumption which ultimately may be found to overestimate risks.

The effect of whole-body exposure in all workers is about 0.2-0.8 excess lifetime cancers/1000 MWe-yr. For the groups with significant lung dose from alpha radiation (mining, milling and fuel fabrication), additional lung cancers will occur from this exposure. These would be about 1.3 excess lung cancers from a year's exposure, about 3X as many as from whole-body exposures in the current cycle. These additional lung cancers indicate that mining and fuel fabrication especially contribute a large portion of the total occupational cancer impact from radiation.

To the 1.5-2.1 excess lifetime cancers arising from annual exposures must be added a component from lung disease and accidents. Preferably, these should be in terms of morbidity, but no data for morbidity exist for uranium miners. About 150 excess deaths from chronic respiratory disease and accidents occurred in the cohort studied by Archer et al. (6), consisting of nearly 4000 underground miners exposed for about 20 years. Scaled to 200 miner-yr/1000 MWe-yr this excess mortality is about 0.4/yr. On the assumption that serious morbidity frequency is at least 8 times mortality, the added health

effects are about 3, 2 from accidents and 1 from respiratory disease. This estimate is based on past practices and may be reduced in the future, but it indicates that injuries and respiratory disease contribute significantly to total morbidity in uranium mining. Morbidity from accidents in the rest of the cycle has been minimal by comparison.

We may summarize the health impacts of the nuclear cycle as in Table 8. The risk estimates are based on current or anticipated normal operations of the nuclear fuel cycle. Occupational on-the-job injury has been included, but no attempt to quantitate risks from accidental releases, such as from criticality accidents or reactor core malfunctions has been included. In my view, the Three Mile Island episode supports the view that if the primary containment vessel remains intact, population exposures from a reactor accident should contribute minimal added risk to the above figures. On the other hand, if a reactor accident led to loss of primary containment, I believe that the political and social consequences would be so severe that the fission nuclear option would be terminated.

TABLE 8

HEALTH IMPACTS OF NUCLEAR FUEL CYCLE
PER 1000 MWE-YR OF POWER PRODUCTION

	Current		Projected for year 2000	
General population	0.028-0.048	Cancer	0.11-0.31*	Cancer
Occupational	1.5-2.1	Cancer	1.4-1.7**	Cancer
	2.0	Trauma	1.0+	Trauma
	0.5	Silicosis	+	
	0.5	Chronic lung disease	0.5	Chronic lung disease
Total Occupational	4.6-5.1		2.9-3.2	
Total	4.6-5.1		3.0-3.5	

*On assumption of recycle of plutonium.
**On assumption whole-body exposures are reduced by one-half, reprocessing not included.
+On assumption injury morbidity is reduced by one-half and silicosis is eliminated.

REFERENCES

1. The Nuclear Industry. U. S. Atomic Energy Commission, WASH 1174-74, 1974.

2. Sources and Effects of Ionizing Radiation Report of the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR), United Nations, New York 1977.

3. Advisory Committee on the Biological Effects of Ionizing Radiation (BEIR Committee), U. S. National Academy of Sciences. Draft Report May, 1979.

4. Radiological Impact Caused by Emissions of Radionuclides into the Air in the United States, Preliminary Report EPA 520/7-79-006, U. S. Environmental Protection Agency, August, 1979.

5. The Health Risk from Inactive Uranium Mill Tailings Piles. Bioeffects Analysis Branch, Criteria and Standards Division, Office of Radiation Programs, U. S. Environmental Protection Agency, Sept. 1979.

6. Archer, V.E., J. D. Gillam, and J. K. Wagoner. Respiratory Disease Mortality Among Uranium Miners. Symposium on Occupational Carcinogenesis, Ann. N.Y. Acad. Sci. 271:280-293, 1976.

SECTION 3

HUMAN HEALTH IMPACTS FROM THE COAL CYCLE

Within the industry providing coal for electric power generation, considerable differences in types of mining, transportation and other factors that can influence health impacts exist throughout the country. As a result one would anticipate that the health impacts of the coal cycle could be substantially influenced by conditions modifying local practices. An example is the extent to which western coal will be used in eastern power plants, because long transportation distances will increase the probability of accidents affecting both workers and the general public. In this presentation I shall consider practices in the ORBES region. In this case, a large proportion of coal used is mined in the ORBES area, although the fraction of western coal used has recently increased because of the high sulfur content of eastern coal. Because the ORBES region has extensive coal deposits, mine mouth power plants are feasible and are currently operating on a limited scale. Furthermore, the mix of transportation modes includes in this area more use of trucks and barges than elsewhere in the U. S.

As in the case of the nuclear cycle, we consider current technology as the basis for projections of health impact, but it is pertinent to note that the use of coal for power has involved an industry of long-standing, with practices rooted in a past when concern for public and worker health was minimal. This fact is in sharp contrast to the development of nuclear power. One result of the long history of the use of coal for power is that the industry is now in transition, with new technology and regulations which are correcting many of the abuses of the past. Because of rapid changes occurring within the past decade, the health impacts that may be discerned in recent years may not be applicable to the current situation, reflecting as they do a legacy from previous conditions no longer present.

Current evaluations of the health impacts of the coal cycle are characterized by controversies which in large part are based on the reliability of forecasts based on past conditions, with differing projections of the future impacts of current practices. In what follows I have considered both sides of these controversial issues, the most important of which involves the occupational health impacts of coal mining and the impact on public health from airborne stack emissions from coal-fired power plants.

Currently somewhat over 100 gigawatts of electric power are generated each year by coal combustion. A gigawatt is 1000 MW, thus the total power

generation is equivalent to about 110 nominal plants, but because many coal-burning plants are quite small the number of actual plants is large.

On the basis that 4,000,000 metric tons (MT) of coal are required per year for 1000 MWe power generation, of the total 1977 coal mining production of about 600 million MT (1), approximately 2/3 was utilized for electric power production. Additional steps in the coal cycle include coal processing, transportation to the power plant, combustion at the plant, and waste disposal. Because figures related to health effects are often given as totals for the entire coal industry, the fraction of 2/3 of coal use for power generation is used wherever applicable.

COAL MINING

Surface mining of coal is now the dominant mode of coal mining, accounting for over 60% of the total production. In the ORBES region the proportion varies considerably by area, with more underground mining in West Virginia and western Pennsylvania. Surface mining is not only more labor-efficient, with less than 30% as many miners required per unit production as is required for underground mining, but it also has less health impacts per miner. Despite economic advantages for surface mining, estimates are that the proportion of coal mined from underground operations will continue about the same as at present, at least into the immediate future in the ORBES area, because of limitations on coal deposits accessible to surface mining and new regulations for such mining. Surface mining is primarily done by the "strip" mining method, in which overburden is removed, the coal taken out, and the overburden is replaced.

Underground mines vary greatly in size, degree of mechanization and techniques of mining. For the large mines, long-wall mining is now becoming very widely applied, but in small mines, sometimes family owned and operated, resort to manual methods is still extensive. These small mines are gradually being forced out of production, in part because of regulatory pressure to prevent accidents and exposure to dust in the mines.

Occupational Health Impacts: Injury

Coal mining has traditionally been one of the most hazardous of all occupations, with frequent episodes of multiple deaths or serious injuries due to cave-ins or explosions. These latter episodes have largely been eliminated by regulations and proper inspection procedures, but the problems of injuries among coal miners remains serious, nevertheless, especially among the smaller mines. In 1977 there were 100 deaths reported from underground mines and 29 from surface mining (1). When these deaths are scaled to production requirements for a nominal power plant, they give 0.7 deaths from underground mining and 0.2 deaths from surface mining, or a total of 0.9 deaths per 1000 MWe per year. A similar calculation for disabling injuries gives about 96/yr, but these are all injuries with lost time greater than an arbitrary limit, without regard to severity or permanent disablement. A

difficult problem, also present in assessing the health impacts of uranium mining, is the extent to which these non-fatal injuries contribute to the total health cost.

Occupational Health Impacts: Disease

Coal workers pneumoconiosis (CWP) is now well-recognized as a clinical entity. With continuing exposure to dust in the mines, the process may progress in a small percent of cases to progressive massive fibrosis, a chronic and disabling disease whose course is independent of continuing exposure. Simple CWP is, on the other hand, not disabling and shows little functional impairment. In addition, Morgan (2) believes that there is a form of industrial bronchitis associated with inhalation of dust, which subsides on removal from exposure. All of these clinical conditions are confounded by cigarette smoking, and the question of whether inhalation of coal dust accelerates or adds to the disease processes associated with smoking has been a source of great debate.

Another complication in assessing the disease impacts of coal mining is the improvement of dust conditions in the mines in recent years. Much of the disease present among miners, especially the more disabling forms, reflects working conditions that are now rare. For this reason the projections of present and future health impacts are difficult; in addition the controversy over the health significance of simple CWP is another source of uncertainty.

For these reasons estimates vary considerably of the new cases of significant lung disease generated per year in a population of 1000 underground and 400 surface miners required for production of coal for a nominal plant. Prevalence rates for CWP must be corrected to take account of the time required for the disease to develop at current exposure levels. If we take the prevalence of CWP as 10% for underground miners (the prevalence of 4% for surface miners is believed to be due to previous underground work), then the prevalence would be 100 cases/nominal plant. On the assumptions that 20 years' exposure are required to develop evidence of the disease, and no more than 20% of the simple CWP cases will progress to functional disability, the new cases per 1000 MWe/yr would be 1.0. This is a relatively crude estimate, perhaps too high if the proportion of cases that progress to massive fibrosis or bronchitis is overestimated for current exposure conditions, and perhaps too low if subsequent evidence supports the view that simple CWP is associated with acceleration of onset of chronic lung disease and eventual disability, at least in smokers.

Health Impacts on General Public

Coal mining has significant effects on populations living nearby, but the long-term health significance of these effects is not well understood. Normal operations result in fugitive dust exposure, especially from strip mining, and siltation of streams. The latter is believed to contribute significantly to flooding, which can have public health significance. In addition, tailings and slag have been used to create earthen dams, which have given way and caused major loss of life. Acid mine drainage in the

coal fields is extensive, but aside from esthetic effects and loss of recreational use of streams, public health impacts have not been demonstrated.

Abandoned underground mines may catch fire and smolder for long periods of time, releasing gases at the surface, especially sulfur dioxide, to local areas. Subsidance from collapse of supports in underground mines also frequently occurs, with substantial effects on property values, at least.

It is not possible to assign a value for health impacts from the above effects created by coal mining, particularly because of the episodic and localized nature of some of them. The potential exists for significant public health impacts from mining on local communities, but many of these conditions reflect past practices which presumably are being corrected, and in any case no quantitative assessment of health risks can be made at this time.

COAL PROCESSING

As mined, coal contains about 20% contamination with rocks and other debris. In addition, mechanical crushing or grinding is necessary for some uses. As a result, about 90% of mined coal undergoes processing of some kind, although the best grades of coal are reserved primarily for industrial uses, and power plants can use lower grades for combustion purposes.

Coal processing is a significant source (about 25% of the total from all mining operations) of acid residues which can leach into water, and is a major source of trace elements in leachates. Moreover, tailings from processing must be handled by various waste disposal methods. Thus processing shares with mining some of the same potential public health impacts, but again there is as yet no basis for providing quantitative estimates of health impacts on the general public.

Data for the period 1972-77 indicate that in coal processing plants, when scaled to coal production for a nominal 1000 MWe plant, the fatality rate for workers is 0.05/yr (mostly from haulage or machinery accidents), and the disabling injury rate is about 5.0/yr. In other words, while coal processing is somewhat less hazardous to workers than strip mining, the health impacts from processing are not negligible.

COAL TRANSPORTATION

Coal transportation is a very significant part of the coal cycle, as transportation of coal by rail accounts for almost one-third of all tonnage of freight transported by rail in the U. S. Rail transportation accounts for about 2/3 of all shipments of coal to power plants in the U. S., but in the ORBES region the proportion is probably somewhat less because of the greater contribution of shipment to ORBES plants by water and by truck, as well as the larger proportion of mine mouth plants which utilize conveyor systems for transport of coal.

Exposure to dusts or contamination of the environment is not a significant factor in coal transportation, thus the health effects arise from injuries. For the general public these injuries are related to railway grade-crossing and switching and yard accidents, and truck collisions on the highways.

Rail Transport

Coal may be transported either mixed with standard freight trains, or as unit trains carrying coal alone. The use of unit trains has been increasing, but thus far there are no statistics which permit separation of injury ratio by type of train. On the basis of ton-miles transported, the proportion of health impacts assigned to coal is relatively high because of the density of coal compared to most freight moved by rail. On a train-mile basis, coal accounts for a lesser proportion of total freight movement. Finally, unit trains for coal may have a lower risk for injury, to the general public at least, because of the lower train velocity commonly present. In calculating the health impacts of rail transportation by coal, I have used the train-mile basis as the best indicator of risk to the public as well as to the railroad workers. The latter assumption may somewhat underestimate the occupational risk because the heavy weight of coal cars may lead to greater probability of injuries associated with switching and coupling operations.

On the assumptions that 4×10^6 MT of coal are required per 1000 MWe, and in the ORBES region there are an average 1.5×10^4 MT carried per train for a round trip distance of 400 miles, the train-miles required per 1000 MWe/yr would be approximately 100,000. From data summarized by Morris (3), the fatality rate to the general public from all rail operations in the U. S. in the period 1966-74 was about 3 per 10^6 train-miles, and the injury rate was about 6 per 10^6 train-miles.

From the above estimates, and on the assumption that in the ORBES region 60% of coal transport is by rail, one would calculate that for the general public there would be about 0.2 fatalities and 0.4 injuries/1000 MWe/yr. Similarly for occupational risks the values would be 0.006 fatalities and 0.5 injuries/1000 MWe/yr.

Truck Transport

Hazard estimates for truck transport are based on vehicle-miles. The same assumption is made as for rail transport that the tonnage transported makes no difference in the accident rate, an assumption that is reasonable because the tons of coal transported per truck (about 20 MT) are similar to other large truck loads. At 20 MT per load, and on the assumptions that in the ORBES region 15% of coal for power consumption is transported by truck and each round trip is 100 miles, the total vehicle-miles per year per nominal plant would be 3×10^6. Statistics for the truck industry for 1973-76 (4) indicate that for truck fleets the fatality rate is 0.009 per 10^6 vehicle-miles. If these values are applied to coal transport, the fatalities would be about 0.03 deaths and 0.1 injuries per 1000 MWe/yr. These deaths and injuries apply both to the general public and to truck drivers, and it is

not possible to apportion them exactly. On the assumption that trucks are more frequently involved in single-vehicle accidents than in multiple-vehicle crashes, but the multiple-vehicle accidents will produce a higher proportion of fatalities or injuries to the general public than to the drivers, we may estimate crudely that about half of the casualties will be distributed to the drivers and half to the general public.

Water Transport

In the ORBES region about 15% of coal is moved by water to power plants, nearly all in barges. Data are available for casualty rates per ton transported by water routes. On the assumptions that the load per barge is the same for coal as for all freight, but that the distance coal is transported to power plants is one-half that for all freight, then rates may be applied to coal transport. From U. S. Department of Transportation data for 1975-76 (4), the casualty rates for freight transport are 0.037 deaths and 0.034 injuries/10^6 ton transported. On this basis and taking account of the shorter transport distance for coal, the fatality and injury rates both would be about 0.01 per 1000 MWe per year. These would apply to the workers only.

We may summarize the health impacts of coal transport in the ORBES region as follows:

		Fatalities per 1000 MWe generated		Injuries per 1000 MWe generated	
Mode of Transport	% Use	General Public	Occupational	General Public	Occupational
Rail	60	0.2	0.006	0.4	0.5
Truck	15	0.015	0.015	0.05	0.05
Water	15	0	0.01	0	0.01

It should be emphasized that while the fatalities can be added, the injuries may vary in severity from one transport mode to the other. The health impacts will be discussed later. It is evident that for the general public rail transport is the principal problem, but for the workers, truck and barge transport also contribute significantly.

COAL-FIRED POWER PLANTS

The possible health effects of emissions from coal-fired power plants have been very extensively studied in some aspects, in others they have not. Examples of the latter are our uncertain knowledge about health effects on the general public from waste leachates, or the lack of occupational health data on plant workers. Because of early interest in possible health effects of airborne sulfur oxides, and because power plants produce in the U. S. a large fraction of the sulfur oxides emitted into the air, very extensive

studies of effects of air pollution, both in animals and in human populations, have been done in the last two decades.

Health Effects on the General Public

Early interest in the effects of sulfur oxides on public health came from the well-known smog episodes in the 1950s in London, England, the Meuse valley in Belgium and in Donora, PA. These episodes, which were associated with prolonged meteorologic temperature inversions and closely packed local concentrations of fixed sources burning soft coal with relatively high sulfur content, established clearly that when air pollution reached high levels sustained for several days, lethal effects on the populations exposed could be demonstrated. The 1952 London episode, as well as subsequent ones a few years later, has been most extensively studied, and from this analysis it is evident that an increased risk of mortality from cardiopulmonary diseases occurred when both suspended particulates and sulfur oxides both exceeded 500 micrograms per meter^3, with the likelihood that transient elevations above 1000 micrograms/m^3 accounted for most of the excess mortality. The potential public health seriousness of exposure of people with varying susceptibilities to air concentrations of soft coal effluents of this amount is generally accepted and has formed an important bench mark in the political process of air pollution control.

Such high concentrations of these pollutants have rarely ever been observed in the U. S., and now with relatively little coal used for home heating and even minimal controls on emissions from coal-burning plants, they are a thing of the past. For this reason, research in the past decade has been directed toward defining the health impacts from lower concentrations of a number of pollutants, arising from coal combustion with particular emphasis on sulfur oxides. Animal studies have generally been of limited value in defining chronic effects in special populations at risk, and thus epidemiologic research has been extensive. A number of conclusions of these studies can now be drawn (see e.g., summaries by Ferris (5) and Higgins (6)).

1. Except for the London studies, research on the specific effects of coal combustion effluents prior to 1970 cannot be relied upon to give any quantitative estimates of risk, primarily because of inadequacies in the air pollution measurements, and also because of confounding effects of cigarette smoking and exposure to indoor pollutants.

2. Recent studies, both experimental and epidemiologic, have shown that concentrations of sulfur dioxide, by itself, in the range of less than 500 micrograms/m^3 have no significant short-term effect on respiratory mechanics, symptoms or disease.

3. If acid derivatives of sulfur dioxide, such as arise from chemical reactions in stack plumes or during long-range transport of effluents, contribute to health effects observed in population studies, their contribution cannot as yet be isolated from the possible influence of other elements in

the suspended particulates present in the airborne mix, such as oxides of nitrogen (largely derived from combustion sources other than power plants), and trace metals.

4. It has been possible to relate acute morbidity and mortality from cardiovascular disease, but not respiratory disease, to local concentrations of total suspended particulates in the range of 300 $\mu g/m^3$ or more (Mazumdar, personal communication), and the recent analysis of New York mortality data by Schimmel (7) suggests that excess total mortality is correlated with suspended particulates, but not sulfur dioxide, in a reasonably dose-related fashion. The fact that in this study respiratory disease mortality is much less strongly correlated with airborne particulates, a phenomenon also observed in the early London studies, strongly supports the view that cardiovascular disease, which accounts for most of the remaining non-cancer deaths, is the principal cause of mortality associated with acute air pollution effects. The biological mechanisms by which inhalation of airborne particulates influences cardiovascular disease mortality remain obscure.

5. The role of airborne irritants in exacerbating asthmatic attacks has received some support from epidemiologic studies, but the particular contribution of effluents from coal-burning is far from clear, especially in view of the large number of irritants in addition to sulfur oxide derivatives that are present in urban air, and the role of indoor pollution in such studies. Some contribution of urban airborne particulates on morbidity and mortality in severely affected asthmatic patients cannot be ruled out, but there is no quantitative basis for assessing risk at this time.

6. From studies of lung cancer in occupational groups, such as coke-oven workers, roofers, and gas-plant workers, exposed to high concentrations of polynuclear aromatic hydrocarbons, it is evident that the concentrations of these combustion products in urban air (about three orders of magnitude lower than the concentrations present in the above occupations) cannot be given any significance in excess rates of lung cancer in urban populations. Weinberg (doctoral thesis, University of Pittsburgh, 1980) has shown that all of the differences in lung cancer rates between census tracts of the Pittsburgh metropolitan area, where considerable variations in coal-derived suspended particulates exist, can be accounted for by differences in cigarette smoking habits. One component of particulate effluents from coal combustion that has not yet been evaluated for a role in urban lung cancer rates is radioactive lead-210 and polonium-210, but at the present time the evidence indicates that there is no association of urban air pollution, including coal-combustion effluents, with excess lung cancer risk.

7. Contributions of sulfur oxide pollution to acid rain, visibility reduction and corrosion of materials are outside the scope of this discussion.

In assessing the public health impacts of power plant stack emissions, therefore, at current levels of pollution we are left with only the relationship of gross airborne particulates to cardiovascular disease mortality as a reasonably firm conclusion at this time. The contribution of sulfur emissions

to this relationship is obscure. Allegations of much more significant health impacts from urban air pollution, which are based on multiple regression techniques, simply display the clear inadequacy of this method for evaluating health effects in complex social groups where major confounding variables such as cigarette smoking cannot be taken into account. Radioactive heavy elements are part of the particulate phase from coal-fired plants, but estimates of risk depend on their route of exposure to human populations. For uranium, thorium and radium isotopes, ingestion from this source is negligible compared to normal soil sources of these elements in food. As mentioned above, however, by the inhalation route particularly, Pb-210 and its alpha-emitting granddaughter Po-210 could contribute to lung cancer risk. This component of the radioactive emissions from coal-fired plants has not yet been evaluated adequately.

My analysis of the recent data indicates that cardiovascular disease mortality (morbidity should reasonably closely correlate with mortality for these diseases) is increased by 5% with sustained exposure to airborne particulates at a concentration of about 350 $\mu g/m^3$. With a proportional extrapolation to higher doses, this figure is in reasonable agreement with the earlier London data. We may now estimate the public health impact of the usual concentrations of urban particulates, and the contribution of power-plants, but only on the assumption that the fraction of urban particulate pollution from coal-fired power plants contributes proportionally to these effects. Until we know more about the mechanisms responsible for this relationship, it is difficult to say whether this assumption leads to over- or under-estimates of the role of power-plant emissions.

To apply the above risk relationship to usual population exposures to suspended particulates which are no more than an annual average of about 100 $\mu g/m^3$ for comparatively heavily polluted urban areas, some dose-response relationship must be assumed. A more difficult problem is presented if one assumes that peak concentrations are the primary factors contributing to mortality, but because the assumption of an effect from sustained exposure leads to higher risk estimates in general, it has been adopted. The linear model can be applied to calculate excess risk by assumption of a background exposure from non-combustion and industrial sources, which I take here to be about 50 $\mu g/m^3$, which is found in rural areas isolated from human activities, and assumption of a value of 50 probably again will slightly overestimate the risk.

Assumption of a straight-line dose-response model must be considered the upper limit case because internal exposure to cells from inhaled particulates would be expected to be under biochemical influences that would reduce the effects of low doses relative to higher doses. In particular, these influences include intestinal and respiratory uptake and excretion, as well as the toxic effects of cell interactions with the offending agents, whatever they are.

Calculations of health effects per 1000 MWe generation may be done as follows:

> The total cardiovascular mortality in the U. S. is about 600,000 per year. Exposure to particulates at levels of 100 $\mu g/m^3$ is estimated to be to no more than 1/2 the urban population, which is 2/3 of the total U. S. population. On the basis of the straight-line dose-response hypothesis, if 5% increased risk occurs at a sustained exposure to 350 $\mu g/m^3$, or 300 $\mu g/m^3$ above background, at 100 $\mu g/m^3$, or 50 $\mu g/m^3$ above background, the increased risk is $\frac{50}{300}$ x 5% = 0.83%. But coal-fired power plants contribute only part of the total urban particulate load, and I assume for the ORBES region 1/5 for this purpose. The maximum total excess cardiovascular mortality contributed per year by coal-burning power plants is therefore:
>
> 600,000 x 1/3 x 0.0083 x 0.2 = 333 deaths
>
> The maximum contribution per 1000 MWe per year, based on current production of 110 GWe/yr from coal plants, is thus 3.0 deaths or potentially lethal incident cases.

This estimate is clearly very uncertain, as the above assumptions attest, but I believe it is a defensible upper limit based on current epidemiologic evidence. The lower limit is zero, because the straight-line estimate of dose response may well not hold at low doses, as defense mechanisms of the body may be able to cope with low-dose exposures without any significant effect on cardiovascular disease.

Although the upper limit estimate of risk to the general public is lower than others have claimed, it is by no means negligible. Indeed, if subsequent work supports a risk estimate close to the upper limit, it is evident that this problem is the principal source of health impacts to the general public from either the coal or nuclear cycles, a conclusion consistent with conventional wisdom in this field.

Occupational Health Impacts

Disease specifically related to occupational exposures in coal-fired plant workers has not been investigated. In conventional modern plants with negative pressure coal feed, exposures to dusts and gases in the workplace should be minimal. The possibility of exposure to fugitive dusts from coal and waste storage exists, but their potential effects are unknown.

With regard to injuries among these workers, Rom (8) reports that for all power plants the fatality rate per 1000 MWe was estimated to be about 0.02 per year and injuries about 1.2 per year. It is not clear whether these rates include transmission linemen; if they did, transmission line injuries would account for a high percentage of the serious injuries estimated for power plant workers (see transmission section 4).

Waste Disposal

Coal-fired power plants generate ash and scrubber sludge that must be disposed of. Although some waste is held temporarily in the plant area, permanent disposal is to sites generally near the plants. Current plans include use of slurry pipelines for transport. Although there is the possibility of leaching of potentially hazardous wastes from storage areas into water supplies, there is as yet no evidence of public health impacts into water systems from power plant or waste effluents (9).

SUMMARY OF HEALTH IMPACTS FROM COAL CYCLE

It is evident from the above discussion that, in contrast to the nuclear cycle, the coal cycle injuries constitute major public health impacts for both the general public and workers. While fatality rates from injuries can be clearly assigned a public health importance, non-fatal injuries are somewhat more difficult to assess in quantitative terms. For example, an injury that is associated with permanent disability, such as quadriplegia, is obviously comparable in its effects on the individuals involved as a fatality may be. In contrast, a broken bone, which may require a long period of loss of work capacity and thus be included as a serious injury, nevertheless may heal completely with no permanent loss of function.

It is a remarkable fact that the ratios of serious lost-time injuries to fatalities among most of the occupational groups discussed above is quite constant at about 80 to 120. But most of these injuries, while significant, produce public health impacts which are substantially less than a fatality or permanently disabling injury. Some adjustment is therefore required to bring these relatively minor injuries into some sort of balance with the more serious ones. On the assumption that occupational injuries are 100 times the fatality rate (a ratio that does not hold for motor vehicle or barge accidents), if one assigns an arbitrary "coefficient of harm" to the range of effects of injury, then it is possible to integrate these and produce a total health impact estimate which is related to the fatality data. In what follows I have used a range of 5 to 10 times the fatality rate as the total non-fatal injury impact equivalent in terms of total health effects to a fatality. It is apparent that on this basis, calculation of health impacts based solely on fatality rates would seriously underestimate the total effect. For those cases where the ratio of injury to fatalities is much lower than 100, I have used either the total injury rate or 5x the fatality rate, whichever is less, to estimate the total health impacts.

On the above assumptions we may summarize the health impacts of the coal cycle:

Health Impacts per 1000 MWe/yr from the Coal Cycle

	Current		Projected to year 2000	
	General Public	Occupational	General Public	Occupational
Mining	None known	5-9 (injury) 0.5 (CWP) 0.5 (Bronchitis)	-	2-5* 0.3* 0.2*
Processing	None known	0.3-0.5 (injury)	-	0.1-0.3*
Transportation	0.7 (injury)	0.1-0.15 (injury)	0.4*	0.1*
Power Plants	0-3.0 (Cardio-vascular disease)	0.1-0.2 (injury)	0-1.5*	0.1
Waste disposal	None known	Negligible	-	-
Total	0.7-3.7	6.5-10.9	0.4-1.9*	2.8-6.0*

*On assumption of a 50% reduction by control techniques.

Comparison of the health impacts from the coal and nuclear cycles leads to the following conclusions:

1. Risks to the general public from the coal cycle are substantially greater (currently by a factor of at least 50) than risks from normal operations of the nuclear fuel cycle, per unit energy produced. This difference is due to public health costs of coal transportation and the possibility of significant health effects from coal-fired power plant stack emissions. If plutonium recycle is adopted, and with improvement in transportation accidents and stack emissions from coal-fired plants, the two cycles become more nearly comparable.

2. Occupational health impacts from the coal cycle are about twice as great for coal use as for the nuclear cycle, per unit energy production, mainly due to the high costs in human health impacts estimated for coal mining. Vigorous efforts should be made to reduce these impacts for both the cycles.

REFERENCES

1. Coal Mine Injuries and Worktime. U.S. Mine Safety and Health Administration Report, 1978 p4.

2. Morgan, W.K.C. Industrial Bronchitis. Brit. J. Ind. Med. 285:35, 1978.

3. Morris, S.C. Health Aspects of Fuel and Waste Transport. ORBES Symposium on Energy and Human Health: Human Costs of Electric Power Generation. U.S. Environmental Protection Agency, Washington, D.C., 1979, pp 260-284.

4. Summary of National Transportation Statistics. U.S. Dept. of Transportation Yearly Reports, Washington, D.C. 1974-77. Also Statistical Abstract of the U.S., 1979.

5. Ferris, B.G., Jr. Health Effects of Exposure to Low Levels of Regulated Air Pollutants: A Critical Review. Journal of the Air Pollution Control Association. 28:5, pp 482-497, 1978.

6. Higgins, I.T.T., Welch, K. and Glassman, J. Epidemiological Studies of Human Health Effects. ORBES Symposium on Energy and Human Health: Human Costs of Electric Power Generation. U. S. Environmental Protection Agency, Washington, D.C., 1979, pp. 260-284.

7. Schimmel, H. Evidence for Possible Acute Health Effects of Ambient Air Pollution from Time Series Analysis. Bull. N.Y. Acad. Sci. 54:1052-1108, 1978.

8. Rom, W.N. Occupational Health Aspects of Fossil-Fuel Electric Power Plants. ORBES Symposium on Energy and Human Health: Human Costs of Electric Power Generation. U.S. Environmental Protection Agency, Washington, D.C., 1979, pp 221-245.

9. Andelman, J.B. Human Exposures to Waterborne Pollutants from Coal-fired Steam Electric Plants. ORBES Symposium on Energy and Human Health: Human Costs of Electric Power Generation. U.S. Environmental Protection Agency Washington, D.C, 1979, pp 195-220.

SECTION 4

POWER TRANSMISSION AND ITS HEALTH EFFECTS

Power transmission is common to all kinds of electric power generation and fuel types. Electricity is transmitted by alternating current at both high and low voltages, high voltages being used for energy transfer for long distances from a generating source to a population center. Lower voltages are used within the population center to distribute electricity to individual consumers. The effects of electricity carried in wires are electric and magnetic fields, and corona effect around the transmission lines. In considering health effects, risks occur to the general public from exposure to the above fields, as well as accidental injury or death from broken or falling transmission lines as a result of wind or ice damage, earthquakes, etc. Occupational hazards also exist from working closer to the lines, not only to the electromagnetic fields but also from direct contact with high-voltage terminals or other low-resistance pathways.

The transmission of electricity from a remote generating source to load centers is accomplished by means of Extremely High Voltage (EHV) lines. EHV levels vary from 230-800 kilovolts (kV). There are advantages to the use of higher voltages within this range. For example, one 765 kV line has the equivalent load carrying capability of five 345 kV lines and its right-of-way requires 145 feet less. With the growth of electric power consumption, the trend has been to use higher voltage transmission. As of January 1, 1970, the circuit miles of AC EHV lines existing in the United States are as in Table 9.

Accurate measurements of the induced currents and power absorbed by a human body from the electromagnetic fields around power lines have been difficult to obtain. Electric field measurements provide the basis for calculating the induced human body current from transmission lines. Practical measurements with a dosimeter show much lower electric field exposure than that calculated on the basis of the maximum field in a work area. Power frequency currents are induced in the body of people who are in the presence of the electric field produced by high voltage transmission lines and substations. Over flat ground the electric field is vertical and uniform over the dimensions of a human being (1). The electric field is not uniform close to non-flat boundaries like vehicles, towers, trees, bushes, and transformers.

The induction field or uniform electric field produced in a human being whose head is 1.76 meters above ground and is walking occasionally through a 20 kV/m field is 5.75 kV/m and corresponds to a space potential of 10 kV

TABLE 9

CIRCUIT MILES OF HIGH VOLTAGE TRANSMISSION LINES IN U.S., 1970

Voltage Levels (kV)	Approx. Circuit Miles
230	1420
287	1020
345	14070
500	6840
700 (765)	66

at the center of the head and the induced current is about 35 microamperes (μA). The body current perception level is about 1000 μA and is larger than the calculated total induced current in either a man on the ground or a lineman. The electric field recognition level and corona (ozone) forming level are 236 Kv/m and 1968 kV/m respectively (2).

Environmental Mortality and Morbidity

Electrical accidents not directly related to linemen working in the transmission industry account for no more than 4% of all fatal accidents and 0.4% of all non-fatal accidents. Statistics for 1962 submitted to the International Symposium on Electrical Accidents showed that of 1118 electrical fatalities, 40% involved high voltages. Of the low voltage fatalities, 43% were industrial, 17% were agricultural, and 39% were domestic. Most HV accidents are occupationally related to electrical generation, transmission, or distribution; while more than half of the low voltage deaths occur during use of portable electrical equipment. Thus the exposure of the general public to injury from transmission line failures is very low.

Reviews of the biological effects of high voltage electric fields have been published. Four studies on short-term exposures to electrical fields involved the medical treatment experience of people living within 25 meters of a transmission line, the experience of farm workers on 18 farms traversed by a 765 kV line, and a physiologic state study involving lineworkers intermittently exposed to intense electric fields during maintenance work. No immediate, acute, or permanent effects were found; however, only 2 of these studies looked at long-term exposure (3).

In contrast, studies on workers exposed to electric fields in Soviet and Spanish high-voltage switchyards report effects such as excitability, headaches, drowsiness, fatigue, and nausea, but these effects may be related more to acoustical noise in the switchyard environment.

No immediate or acute effects were noted in laboratory animals exposed to electric fields. A small reduction in weight gain was observed in the progeny of mice exposed to intense fields (100 kV/m), and there was evidence of behavioral changes. Cell-mediated immunity changes were reported for rodents exposed to intense fields up to 100 kV/m (3).

Corona Effects

Corona is an electrical discharge occurring at the surface of a transmission line conductor when the electric field intensity at the surface of the conductor exceeds the breakdown strength of the air. Corona effects associated with transmission lines could generate ozone and nitrogen oxides. According to one study, emissions due to a 765 kV line would result in concentrations of 2 to 3 ppb of ozone not to be exceeded more than one hour per year. Ozone is believed to produce adverse health effects at about 100 ppb, but the effects of exposure to 2-3 ppb over a long period of time are

unknown. On a linear dose-response hypothesis, the effects should be negligible. Nitrogen oxide would be at about the same concentration as ozone, also considerably below concentrations at which any effects would occur.

On the basis of the above data, the conclusion seems warranted that health effects to the general public from electric transmission are not yet demonstrated, and in any case are likely to be negligible in relation to the hazards from parts of the electric power generating system and the fuel cycles on which it depends.

Occupational Risks: Injury

Persons employed in electrical generation, transmission, and distribution are exposed to additional hazards as well as the normal work exposure of falls, lifting, etc. Many electrical accidents involve contact with A.C. 50 or 60 Hertz currents and most high-voltage accidents are occupationally related. Since linemen working on high and low voltage distribution equipment are working at elevated height, many electrical accidents cause severe injuries as a shock causes the body to involuntarily react and the worker falls from a pylon.

The various effects of accidental contact with electric currents include asphyxia from respiratory arrest caused when the electric current passes through the head. Very few of these accidents are fatal, since the head is not a common path for the current to take. Ventricular fibrillation occurs less when the current is over several amperes, so victims of HV accidents often survive, though severely burned. Electric burns and flash burns occur from HV accidents. Finally, secondary effects, such as angina electrica, are clinically undistinguishable from angina pectoris, but clear up after a few months; chromoproteinuria and renal function disturbances and various neurological sequelae have been described (4).

Analysis of the injury and illness rates for the electrical industry in 1974 showed no difference from industrial experience as a whole, but in 1976 the death rate for all industries was 14 per 100,000 workers, while for Transportation and Public Utilities was 31 per 100,000. How much of this excess was related to transmission-line workers is not clear.

In a study done by the Society of Actuaries in 1967, mortality ratios and extra deaths were calculated for occupations considered substandard by most companies for either life insurance or accidental death benefits; other occupations were included if there was reason to expect mortality somewhat above that for standard risks. Based on U. S. Bureau of Labor Statistics data for 1963 on-job occupational deaths, the on-job accidental rate for employed persons is 0.15 per 1000. The Standardized Mortality Ratio for violent deaths among transmission workers was 243 for non-climbing workers, but 425 for climbing workers. Thus the excess mortality for climbing linemen is about 0.27 per 1000. On the assumption that serious injuries could be four times the mortality, total injury would be about 1 per 1000 workers/yr.

To determine possible long-term health effects of electric fields, a medical follow-up study of high voltage linemen working in AC electric fields over a period of nine years was carried out. No disease states which could be in any way related to HV line exposure were found. Examinations consisted of complete medical history, physical, hemotological and blood chemistry studies, thyroid test, kidney and liver function tests, electrocardiograms, electroencephelograms, hearing test, x-rays, sperm count, and psychiatric evaluation. This is relatively short exposure, however, and the end points were non-specific.

It is difficult to convert occupational risks from injuries to a nominal plant requirement, but from Bureau of Labor Statistics about 100 line workers are required per 1000 MWe generation plant. On this basis, the risk per 1000 MWe/yr is about 0.1 serious injury per year per nominal plant.

REFERENCES

1. Deno, D. W. Currents Induced in the Human Body by High Voltage Transmission Line Electric Fields--Measurement and Calculation of Distribution of Dose. IEEE Transactions on Power Apparatus and Systems. Vol PAS-96 No. 5 Sept/Oct 1977, pp 1517-1527.

2. Spiegel, R. J. High Voltage Electric Field Coupling to Humans Using Moment Method Techniques. Biological Effects of Electromagnetic Waves. Vol. II HEW 77-8011, U.S. Govt Printing Off., Washington, DC, 1975.

3. Task Force Report, National Institute of Environmental Health Sciences. Env. Health Perspectives 20:October, 1977.

4. Electricity--physiology, pathology. In Encyclopedia of Occupational Health and Safety, Vol I, Int. Labor Org. Geneva, 1971, pp 449-451.